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# Health Risk Comparison between Groundwater Transport Models and Field Data

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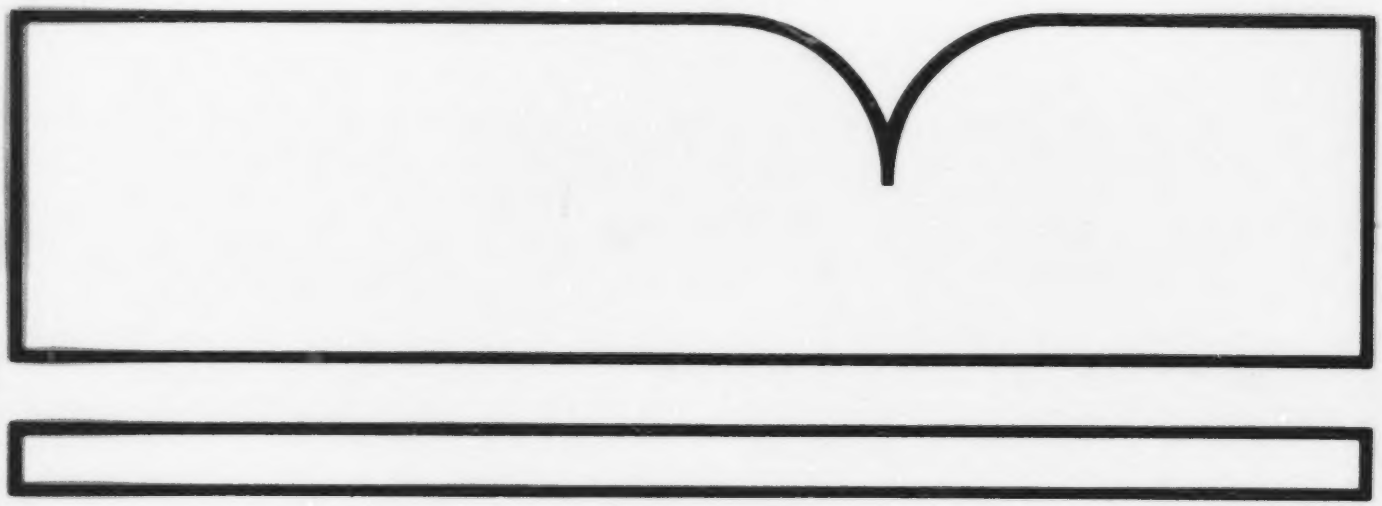

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16. ABSTRACT

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# Health Risk Comparison Between Groundwater Transport Models and Field Data

**The potential of ground water contamination is one of the major concerns over land disposal of hazardous waste. Risk assessment requires information on groundwater concentrations of contaminants at the exposure location. Results are presented of case studies comparing health risk assessment and plume delineation based on state-of-the-modeling and monitoring data.**

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Proper management of solid and hazardous waste should provide acceptable levels of human exposure to toxic waste constituents released from disposal, treatment and storage facilities into ground water. Frequent findings of ground water contamination underscore the ability to conduct an adequate assessment of migration potentials of contaminants as well as their harmful effects on human health as a consequence of drinking the contaminated water. To overcome the weakness in the present state-of-the-art technique for conducting such assessment, current regulatory approaches to waste management tend to emphasize control technology for waste containment, monitoring to detect contamination, and the alternate method of setting contaminant limitations based on consideration of short and long-term human health effects. There is a more recent Congressional mandate (such as 1984 reauthorization of the Resource Conservation and Recovery Act) totally devoted to waste characterization for the purpose of banning toxic waste from disposal.

Despite these different conceptual approaches to waste management, the basic need for predicting contaminant migration has not diminished, but seems to be ever growing. This is because exposure to contaminants that have migrated or have potential to migrate needs to be assessed for sound risk assessment, regardless of the conceptual approaches used.

This paper will present the methodologies that have been used in conducting exposure assessments for sites where ground water monitoring data show contamination, and also where no field data are available. Ground water monitoring data and transport models were used to determine the levels of contaminant concentrations in ground water. Models used range from computerized models to simple analytical models [*3, 4, 6, 7, 8, 9, 11, 13*]. The paper also presents the results of health risk assessment for protection of human health from carcinogenic and noncarcinogenic effects based on the exposure assessments.

## GENERAL METHODOLOGY

Assessing exposure to contaminants in ground water impacted by waste facilities is a stepwise process which may involve the following steps: (1) site characterization for collecting relevant site-specific information, (2) collection of available monitoring data, and (3) estimating concentrations at the points of exposure.

Monitoring data from the point of exposure should be used wherever available for estimating the potential intakes of waste constituents. Since the concentrations in the plume will be steadily changing, the concentrations monitored at a well location are transient in nature. If the site is releasing contaminants continuously, the plume dimensions will grow, thereby increasing the concentrations at the exposure point until steady-state concentrations are attained.

Since monitoring data could be a measurement of dynamically changing plume dimensions, it is appropriate to ascertain that the data represent steady state conditions at the time of sampling. Unacceptable exposures, as indicated by the monitoring data, should trigger further investigation or some sort of corrective action for steadily growing plumes, unless remedial measures have been completed at the site. The monitored concentrations below an acceptable exposure level do not always guarantee future safety in protecting public health and the environment from exposure to contaminants being released, until no further growth in plume dimensions can be ascertained.

Monitoring data obtained from a facility operating over an extended period of time may correspond to steady-state concentrations. In such cases, any significant increase in concentration in the future may be unlikely. Models can be used to simulate the conditions existing in the ground water environment, estimating transient and steady state concentrations which may be attained.

The concentration at an exposure point determined from monitoring data, or calculated from transport modeling can be used to estimate the health effects on a short- or long-term basis. A compilation of health effect data is available which are derived from the weight of evidence on carcinogenic effects, and short-term and chronic noncarcinogenic effects.

One type of health effect model expresses the carcinogenic effect as a linear relationship between dose and effect, called the potency factor $K(mg/kg \cdot day)^{-1}$, while noncarcinogenic effect data are used to derive acceptable daily intake (ADI, mg/day) dose below which no adverse noncarcinogenic health effects can be observed. Because of difference in body weight between an adult



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and a child, a 70 kg adult will require a higher ADI than a 10 kg child.

The risk(R) associated with the intake of contaminants in normal 2 L/day of drinking water over a lifetime can be obtained by

$$R = 1 - e^{-kD}$$

where $D$ is the daily intake of a contaminant present in drinking water per unit weight of human body (mg/kg · day). When the cancer risk is less than $10^{-3}$, this formula can be approximated by $R = K \cdot D$.

## ASSESSMENT USING MONITORING DATA

The data obtained from ground water monitoring wells can be effectively used to conduct health risk assessment. The concentration level will be different depending upon the location of monitoring wells in relation to waste treatment, storage and disposal facilities where contaminant migration occurs. The concentration should be evaluated to determine if the level is acceptable to prevent the carcinogenic and noncarcinogenic health effects.

As indicated previously, the concentration variation is a dynamic process being affected by hydrogeologic conditions under which contaminant transport occurs, and chemical and physical properties of the contaminants. The concentrations are also changing in time and space unless steady state conditions are reached. This consideration should be important in making conclusions on health effect consequences based on monitoring data. The results based on transport modeling will be helpful in determining the degree of variation in health risk assessment over a time period of interest.

## ASSESSMENT USING MATHEMATICAL MODELING

Although the usefulness of mathematical modeling as an important tool in predicting contaminant migration in ground water has long been recognized, reluctance on its use relates to the depth of understanding of fundamental processes that control fate and transport, and legimate confusion stemming from "too many" available models that can be found in the literature.

Before resorting to mathematical modeling, it is critical to define the objective of using it. This process is important in selecting appropriate models among various models available in the literature, as well as in defining the significance of assigned values for chemical, physical and hydrogeologic parameters that affect transport processes. The objective could be one or more of the following:

1. To compare monitoring data with calculated results.
2. To estimate the dynamic nature of monitored plumes.
3. To estimate concentration levels in the absence of monitoring data.
4. To compare the results of different models.
5. To compare the results based on different parameter values.

Whatever the resulting solutions of the models may be, the starting equation describing the fate and transport of contaminant in ground water is obtained by writing a material balance around an infinitesimal flow element in the aquifer. For steady flow through a uniform aquifer medium, the partial differential equation takes the form

$$R_d \frac{\partial C}{\partial t} = D_x \frac{\partial^2 C}{\partial x^2} + D_y \frac{\partial^2 C}{\partial y^2} + D_z \frac{\partial^2 C}{\partial z^2} - V \frac{\partial C}{\partial x} - kR_d C \quad (1)$$

where C = concentration of a contaminant at a ground water location $x$(cm), $y$(cm) and $z$(cm), g/cm$^3$; $V$ = seepage velocity, cm/s, which can be obtained by dividing Darcy's velocity by porosity $\epsilon$; $R_d$ = retardation factor: $D_x$, $D_y$, $D_z$ = effective diffusion coefficients in $x$, $y$ and $z$ directions, respectively, cm$^2$/s, which are normally expressed by the relationships $D_x = \alpha_x V$, $D_y = \alpha_y V$ and $D_z = \alpha_z V$, where $\alpha$ represents dispersivity in the respective direction; $t$ = time in sec., and $k$ = biodegradation constant, 1/s.

Different models result not only because Equation (1) is solved analytically or numerically, but also because it is solved under the assumption of one, two or three dimensional transport problems depending upon the simplication needed. There are also numerous computer codes, some being proprietary, simulating ground water transport by numerically solving Equation (1) coupled with flow equations and appropriate boundary conditions [*6, 7, 9*]. A computer code in the public domain makes use of a technique tracking particles initially placed in grid cells [*9*]. Also many of the analytical solutions have been documented [*4, 8, 13, 15*].

Since one or two dimensional solutions are often unrealistic for three dimensional aquifer problems, a few three dimensional solutions which can be easily used in exposure assessment are presented herein. Because of space limitation, the technique of numerical solution is not explained in this paper.

The analytical solution to the three dimensional transport model Equation (1) under the condition of a continuous point source discharge is taken from Turner's solution [*12*], and takes the following form:

$$C(x,y,z,t) = \frac{C_o Q}{4\pi\epsilon R\sqrt{D_y D_z}} e^{\frac{Vx}{2D_x}} \left\{ e^{\frac{UR}{2D_x}} \operatorname{erfc}\left( \frac{Ut + R_d R}{\sqrt{4R_d D_x t}} \right) + e^{-\frac{UR}{2D_x}} \operatorname{erfc}\left( \frac{R_d R - Ut}{\sqrt{4R_d D_x t}} \right) \right\} \quad (2)$$

where

$$R^2 = x^2 + \frac{D_x}{D_y} y^2 + \frac{D_x}{D_z} z^2; \; U = V\left( 1 + \frac{4D_x R_d k}{V^2} \right)^{1/2}$$

The expression for steady state concentration can be easily obtained from Equation (2) by letting $t \to \infty$, and using erf (0) = 0.

The release from land disposal facilities does not in general approximate the point source assumption, unless it occurs in a form of leakage through a punctured hole in containing liners. Leakage from underground storage tanks may often resemble point source release. Since the release from land disposal facilities impacts ground water from an area on the top of flowing water rather than in a form of point source release, Equation (1) can be solved with the following boundary and initial conditions:

1. $C = C_b$ (background concentration) at $t = 0$, $z > 0$, and all $x$ and $y$

2. $\dfrac{\partial C}{\partial z} = 0$ at $z = H$

3. $VC - \epsilon Dz \dfrac{\partial C}{\partial z} = M$ at $z = 0$,

$$-\frac{A}{2} \leq x \leq \frac{A}{2}, \; -\frac{B}{2} \leq y \leq \frac{B}{2}$$

where $H$ = aquifer depth, $M$ = rate of contaminant release per unit area, g/cm$^2$ · s, and $A$ and $B$ are the dimensions of a disposal facility in the direction and cross-direction of ground water flow. See Figure 1 for an idealized sketch showing a plane source on the groundwater surface, and waste cells from which leakage occurs.

The solution to Equation (1) with the three boundary and initial conditions given above are

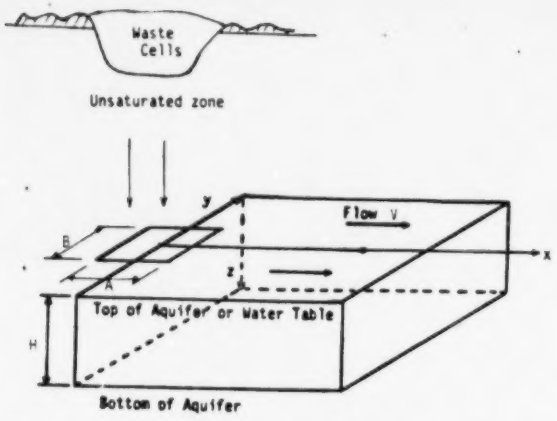


Figure 1. Idealized sketch of a horizontal plane source and waste cell arrangement approximating the source.

$$C(x,y,z,t) = \frac{C_o Q}{4\epsilon R_d A \cdot B} \int_{t'=0}^{t'=t_d} \frac{1}{\sqrt{\pi D_z (t-t')}} \operatorname{erf}\left(\frac{H}{\sqrt{4D_z^*(t-t')}}\right) \exp\left[-\frac{z^2}{4D_z^*(t-t')} - k(t-t')\right] \cdot F\, dt' \quad (3)$$

where

$$F = \left[\operatorname{erf}\left(\frac{x - V^*(t-t') + A/2}{\sqrt{4D_x^*(t-t')}}\right) - \operatorname{erf}\left(\frac{x - V^*(t-t') - A/2}{\sqrt{4D_x^*(t-t')}}\right)\right] \left[\operatorname{erf}\left(\frac{y + B/2}{\sqrt{4Dy^*(t-t')}}\right) - \operatorname{erf}\left(\frac{y - B/2}{\sqrt{4Dy^*(t-t')}}\right)\right]$$

$V^* = V/R_d$; $D_x^* = D_x/R_d$; $D_y^*, D_z^* = D_z/R_d$; $t_d$ = length of source discharge time in seconds; $H$ = depth of aquifer in cm; and the other symbols have the same meaning as in Equation (1). Equation (3) can also be integrated over the aquifer depth to obtain the vertically averaged concentrations in the aquifer over its depth.

The integration in Equation (3) with respect to release time $t_d$ is necessary because contaminants being released are assumed to reach ground water continuously. The integration should be performed numerically from the start to cessation of contaminant release to ground water because no closed form integration is possible.

The analytical or numerical solutions can be used to estimate the concentration level at the point of exposure or to establish plume isopleths under dynamic or steady state conditions. Health risk assessment can be performed based on the calculated results or monitoring data.

## EXAMPLE ASSESSMENT

### Use of Monitoring Data

After an extensive search for candidate sites with good plume delineation, five sites were selected for health risk assessment evaluation. The criteria for selection included the analysis of ground water contaminants for which applicable unit risk (UCR) and ADI values were available. The sites selected after this initial screening were also subjected to modeling if adequate site characterization data were available for determination of transient behavior of the plume and steady-state concentrations. The concentration values were determined in the immediate vicinity of the facility and at a distance away from the source, within the property boundary.

The sites extensively monitored by the U.S. Geological Survey (USGS) were also investigated for possible use in the health risk assessment. Out of about seven sites investigated, only two sites have been selected for evaluation. This determination was made because the other sites either do not have ground water contaminants for which the UCR or ADI values have been determined, or the plume delineation is not adequate.

Table 1 summarizes an evaluation of exposure to contaminants monitored at the waste boundary and approximately 100 feet away from the migration side of the waste boundary. Alternate waste management boundaries other than the 100 feet distance or the waste boundary can be chosen as favored by the availability of monitoring data. EPA's UCR values given in units of $(mg/kg \cdot day)^{-1}$ are obtained mostly from the animal dose-response experiments that are used in determining the risks associated with ex-

TABLE 1. SUMMARY OF HEALTH RISK EVALUATION FOR SITES WITH GOOD MONITORING DATA

| Site No. | Contaminants | Concentration (ug/L) at 100 ft from Waste Boundary | Concentration (ug/L) at Waste Boundary | UCR[c] $(mg/kg \cdot day)^{-1}$ | ADI[f] (mg/day) | Upper Bound Cancer Risk at 100 ft from Waste Boundary | Exposure Ratio for Noncarcinogens (Exposure/ADI) 100 ft from Waste Boundary | Exposure Ratio for Noncarcinogens (Exposure/ADI) at Waste Boundary |
|---|---|---|---|---|---|---|---|---|
| 1[a] | Naphthalene | 650 | 1630 | | 18 | | 0.07 | 0.18 |
| | PHA | 1170 | 1800 | | 18 | | 0.13 | 0.2 |
| 2[b] | Chromium (Hexavalent | 24 | | 41 | | 0.028 | | |
| | Cadmium | 6900 | | 7.8 | | 0.785[c] | | |
| 3 | Arsenic | 623000 | | 15 | | 1 | | |
| | Phenol | | 60 | | 7 | | | |
| 4 | 1,2-Dichlorobenzene | | 16000 | | 6.3 | | | 5.1[d] |
| | Chlorobenzene | | 1100 | | 1 | | | 2.2 |
| | Toluene | | 2300 | | 30 | | | 0.15 |
| | Xylene | | 1900 | | 160 | | | 0.02 |
| 5. | Phenol | 8 | 28 | | 7 | | 0.002 | 0.008 |
| | Pentachlorophenol | 57 | 3400 | | 2.1 | | 0.054 | 3.2 |

a: Data from Reference 1 and private communication with Bedient.
b: Data from South Farmingdale-Massapequa, N.Y. are used for evaluation (10).
c: The risk calculated by $R = 1 - \exp(-7.8(6900/1000)(2)(1/70)) = 0.785$ where a daily water intake rate of 2 L/day, and the weight of a human body of 70 kg are assumed.
d: Exposure/ADI = (16 mg/L · 2 L/day)/6.3 mg/day = 5.1
e: The source of data is Reference (16).
f: The source of data is Reference (17).

posure to carcinogenic contaminants. There is a column in the table, showing the estimated carcinogenic risk for drinking 2 L/day of water contaminated at the corresponding concentrations. According to the definition of UCR, the risk level shown represents the upper bound estimate. An upper bound estimate of risk of 0.028, for example, means that upon lifetime exposure to 2 liters of water per day, a person experiences an increased maximum risk of developing cancer in a probability of 28 in 1000. This level of risk used in this example represents very unacceptable risk, considering that current EPA's strategy is more in line with keeping the risk level at in between $10^{-4} - 10^{-7}$ with a particular level to be based on site-specific characteristics including population distribution.

#### Use of Mathematical Models

Three sites with partially documented and undocumented contaminant plumes were selected for modeling. The movement of a contaminant plume was simulated with a two-dimensional finite element code. The three dimensional analytical models are also employed for comparison purposes. The comparison will be presented later. For the example simulation, the amount of leachate entering ground water was set at 1.0 mg/L of a contaminant, which forms one of the boundary conditions. Site characterization data were collected for the three sites. One of the criteria for selection was the presence of tight geologic formations affecting leachate and ground water flow. This is because geologic formations of the sites with monitoring data showing contamination are relatively all loose with high values of hydraulic conductivity.

Site 6 is a landfill located in northeast glaciated terrain. The cross-section of the site shows fill, silt, clay, loam, and dolomite materials in layers. The simulation for this site was performed for both high- and low-permeability scenarios. Site 7 is a landfill covering approximately 300 acres, which has been in operation since 1977. The facility is underlain by chalk. Two types of clay were identified at the site. Site 8 is located in the Basin and Range physiographic province, and is characterized by a thick, unsaturated layer and infrequent precipitation. Although the site appears to be in a zero recharge zone, the ground water generally flows at the rate of 2 to 4 ft/day within the aquifer.

The principal model input parameters in this simulation were hydraulic conductivity (or seepage velocity), size of the facility, and duration of the facility's operation. Additional information was obtained from the water table contour maps and from characterization data relating to the geologic setting of the sites.

Table 2 is a summary of some of the results obtained from the numerical modeling of the three sites. The hydraulic conductivity rate used for site 6 was obtained from existing data, and ranges from $10^{-5}$ to 8.5 ft/day. The hydraulic conductivity for site 7 ranges from $1.7 \times 10^{-4}$ to $1.7 \times 10^{-5}$ ft/day; and that for site 8 is 8.7 ft/day. The longitudinal dispersivity values used for Sites 6, 7, and 8 are 300 cm, 450 cm, and 75 cm, respectively. The results shown in Table 2 are based on the lateral dispersivity, which is 1/10 of the longitudinal dispersivity. The retardation factor used is 2 for all sites. Grid spacing was used to divide the study area into 288, 462, and 1000 elements, respectively.

Sites 7 and 8 were selected from the facilities for which site characterization information was sufficient for numerical modeling, and which also had low hydraulic conductivity. For this reason, the time-of-travel, defined as the time required for water to travel the 100-ft distance from the edge of the waste discharge area along a ground water flowline, is very large for the sites represented in Table 2. The results indicate that for site 6, the plume at the 100-ft distance closely approximates the steady state concentrations in about 100 years, but for sites 7 and 8, the concentrations are seen to be far from reaching steady-state values in 100 years.

The last column in Table 2 tabulates the risks associated with drinking ground water containing the contaminant at a 100-ft distance under steady state conditions. The table indicates that, when the conditions are such that steady-state is very slowly reached, the exposure assessment based on estimation or monitoring data in the early stage of plume development will lead to an inadequate assessment for the purposes of protecting public health in the future. In computing the upper bound estimates of risk from the data in Table 2, a UCR value of 0.052 $(mg/kg \cdot day)^{-1}$, which is a value for benzene, was used.

Additional computer runs were made for Site 7 using longitudinal dispersivity values of 150 cm and 1,500 cm, and the transverse dispersivities at 10% of the longitudinal dispersivity as some have suggested. The results showed that higher longitudinal dispersivities resulted in higher concentrations at each point during unsteady-state plume development. The simulation was run for up to 10,000 years. The trend indicates that the steady state concentrations increase at greater longitudinal dispersivities.

## LIMITATIONS OF ASSESSMENT METHODS

The use of monitoring data would render the results of health risk assessment more factual than the use of mathematical modeling because actual data would be less subject to uncertainty normally considered present in the estimation. However, the drawbacks associated with the use of monitoring include: 1) Protection of future health risks can not be ascertained if the assessment is solely based on monitoring data, since the plume behavior is a dynamic process until it reaches a steady state condition; 2) The wide-ranging variability in ground water monitoring data may result in a large confidence interval, making it difficult to distinguish the increase in concentration from the background, and hence to determine the extent of contamination; and 3) Extensive monitoring is costly compared with the effective use of modeling technique in combination with monitoring effort.

The uncertainty problem associated with the use of monitoring data is not necessarily resolved by the use of modeling. This is not only because models have variations in describing the fate and transport process, but also because there are a number of chemical and physical parameters needed in the modeling process, that are poorly

TABLE 2. RESULTS OF NUMERICAL MODELING FOR PARTIALLY DOCUMENTED AND UNDOCUMENTED PLUMES

| | Concentrations (mg/L) at 100 ft | | | Time (yrs) | | Risks for steady concentrations |
|---|---|---|---|---|---|---|
| | at 20 yrs | at 100 yrs | steady-state | to reach steady-state | of travel | |
| Site 6 | 0.38 | 0.5 | 0.53 | 1,200 | 1-10 | $8 \times 10^{-4}$ |
| Site 7 | | 0.023 | 0.69 | >10,000 | 1,200 | $1.0 \times 10^{-3}$ |
| Site 8 | | $1.5 \times 10^{-10}$ | 1 | >14,000 | 7,000 | $1.5 \times 10^{-3}$ |

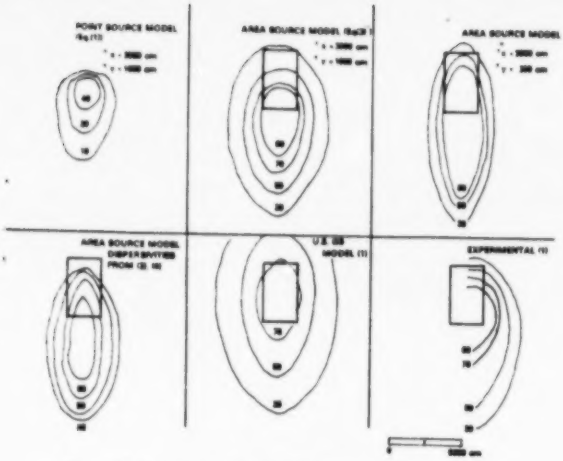

Figure 2. Comparison of plume concentrations (mg/l).

defined. To illustrate the point, analytical and numerical solutions are applied to draw plume isopleths for the identical transport problem. The numerical model used is the computer code developed by the U.S. Geological Survey (USGS), which is available for use to the public [9] Figure 2 compares their results obtained using different dispersivity correlations reported in the literature [*2, 5, 11, 14*] with plume data as found for a creosote disposal site in Conroe, Texas [*1*].

The principal parameter values used in the simulation were as follows: the average seepage velocity at $9.5 \times 10^{-8}$ cm/s; the source width and length at 3,048 cm and 6,098 cm, respectively, lasting for 7,300 days (1952-1972) with the concentration evaluated at $t$ = 10,950 days (1952-1982); aquifer depth at 300 cm; the contaminant concentration $C = 7.5 \times 10^{-5}$ g/cm³ in leachate flowing at a rate of 28.9° cm³/s; and dispersivities. Since the dispersivity values were the most elusive of all of the input parameters, several dispersivity correlations were tried. These included both fixed values and scale-dependent values which become asymtotic at great distances from the source [*1, 2, 8*]. The lateral dispersivity calculated at 1/10 and 1/3 of the longitudinal value was tried.

In this particular example, the use of $\alpha_x$ = 3000 and $\alpha_T$ = 1,000 most closely matches the experimental data. The lateral dispersivity at 1/10 of longitudinal dispersivity was used in the USGS model. The use of the same ratio (1/10) for lateral dispersivity in the area source analytical model [Equation (3)] results in plumes that are somewhat more elongated. The reason for this discrepancy cannot be explained at this time. The use of scale-dependent dispersivities in the area source model results in plume isopleths showing higher concentrations at the location immediately downstream from the source than in the groundwater directly below the source. The concentrations shown for the point source model are those at a depth of 100 cm below the water table. Applicability of the dispersivity ratio used for this example to other hydrogeologic settings is not known.

Correlations for dispersivity by Guven et al., Gelhar et al., Yeh, and Hamilton et al. were applied for comparative purposes [*2, 5, 11, 13*]. As can be noted, difficulty in defining dispersivities makes the sophisticated mathematical exercise often than not fruitless in obtaining reliable estimated results.

## CONCLUSIONS

Techniques of performing assessments of health risk from intake of ground water impacted by waste disposal facilities are presented. The assessment consists of using monitored plume data, fate and transport modeling, or combination of both for exposure assessment, and of using health effect data for carcinogens and noncarcinogens for health risk assessment.

Despite uncertain parameters, particularly values for dispersivities, the fate and transport models are an important tool in supporting the health risk assessment which is based on monitoring data or conducted in the absence of such data. Though there are many analytical and numerical simulation techniques available at present, the reliability of such models cannot be adequately tested because of the inability to define the dispersion parameter for model validation.

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17. U.S. EPA, (May 1984) Summary of Current Oral Acceptable Daily Intakes (ADIs) for Systemic Toxicants, Environmental Criteria and Assessment Office, Cincinnati, Ohio. (The document cautions not be cited or quoted. Hence, not all of the ADI values may reflect current EPA policy.)



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# CORRIGENDUM

Reference 17, U.S. EPA (May 1984) Summary of Current Oral Acceptable Daily Intakes (ADIs) for Systemic Toxicants, was never published and has been superseded by:

U.S. Environmental Protection Agency, 1985. Reference Values for Risk Assessment. Prepared by the Office of Health and Environmental Assessment, Environmental Criteria and Assessment Office, Cincinnati, OH for the Office of Solid Waste, Washington, DC. The document is in the Resource Conservation and Recovery Act (RCRA) Docket.

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